

Supplemental Appendix 1. Geomorphology and other abiotic drivers of ecosystem functioning in drylands

Geomorphology, a significant natural driver of ecosystem function and structure, controls the type and distribution of different landscape features, and the quality of surface soils (e.g. soil texture), creating characteristic spatial elements of ecosystem structure, and modulating ecosystem process rates both within patches and across the larger ecosystem. Vegetation cover in drylands is characterized by a mosaic of discrete plant patches separated by a matrix of bare ground and/or biocrusts dominated by lichens, mosses and cyanobacteria (Valentin et al. 1999, Whitford 2002, Belnap 2006, Maestre et al. 2011). For example, soil-water-vegetation interactions are pronounced in drylands because water is redistributed from bare or biocrust-dominated areas, which typically have finer soil textures and lower infiltration rates, to plant patches in a process that sustains vegetation growth (Noy Meir 1973, Puigdefábregas et al. 1999, Eldridge et al. 2012, Bhark & Small 2003). This process promotes the concentration of water, nutrients and biological activity within plant patches, forming “fertile islands” (García-Moya and McNeil 1970, Garner & Steinberger 1998, Schlesinger & Pilmanis 1998, Reynolds et al. 1999, Aguiar & Sala 1999, Eldridge et al. 2015). Fertile islands are further maintained by positive feedback processes of organic matter accumulation and recycling within the patch, as well as by higher levels of biological activity (Thompson et al. 2005, Méndez et al. 2008, Bonanomi et al. 2011, Morris et al. 2013, Stahlheber & D’Antonio 2014). Fertile islands become preferred habitat for a range of biota, many that have a disproportionate effect on ecosystem functions (e.g. ecosystem engineers; Jones et al. 1997), with positive feedback effects on soil and nutrient processes and vegetation community structure. Engineering by terrestrial vertebrates and invertebrates can directly or indirectly influence processes such as litter decomposition, nutrient cycling and water flows (Davidson et al. 2008). For example, vertebrates have been shown to modify plant species

composition by altering soil nutrient relationships (Mun & Whitford 1990), can disperse or consume fruit or seed (Murphy et al. 2005), or change the dominant plant growth form by grazing and browsing, thereby altering soil seed banks (Chew 1974, Kerley & Whitford 2000, Burggraaf-van Nierop & van der Meijden 1984). Many of these impacts produce flow-on effects to other organisms (Tardiff & Stanford 1998, Davidson et al. 2008).

Geomorphological processes such as weathering and sediment transport also exert a strong influence on soil properties. Soil texture also influences net primary productivity by affecting soil hydraulic behaviour and soil water holding capacity (Noy-Meir 1979, Dodd et al. 2002), and has been shown to influence faunal communities across a range of trophic levels through its effects on vegetation composition and distribution (Woinarski et al. 1999, Eldridge & Whitford 2014).

Supplemental Appendix 2. Beyond plants: the importance of multi-trophic approaches

There is a growing body of studies showing that interactions between plants, biocrusts and soil microorganisms are the norm in dryland ecosystems, and are major determinants of their functioning (Green et al. 2008, García-Palacios et al. 2011, Jing et al. 2015, Moore et al. 2015, Delgado-Baquerizo et al. 2016a). Available evidence suggest that, due to their simultaneous contribution to ecosystem functioning and their different response to environmental disturbances, plants, biocrusts and soil microorganisms can importantly increase the *response diversity* of dryland ecosystems (i.e., species that drive the same function but respond differently to the environment can insure ecosystem functioning under a broader range of environmental conditions and enhance ecosystem resilience, Yachi & Loureau 1999, Karp et al. 2011). However, no experiment to date has performed crossed removals or diversity alterations of plants, biocrusts and soil microorganisms to investigate

their functional redundancy, and their role on ecosystem resilience (but see Rodríguez-Caballero et al. 2012 for an example using a biocrust-removal experiment).

The diversity or composition of other trophic groups, such as herbivores and predators, is known to interact with the diversity of autotrophs or detritivore organisms to determine ecosystem functioning in a wide variety of ecosystems (Naeem et al. 1994, Petchey et al. 1999, Worm & Duffy 2003), and drylands are no exception to this. A large body of literature documents how the exclusion of herbivores or predators affect plant communities, biocrusts and ecosystem functioning in drylands (reviewed in Meserve et al. 2015, Eldridge et al. 2016). For example, different herbivores show contrasting behaviors to avoid predation, which produces strong variation in their effects on plant communities and ecosystem functioning (e.g., Riginos & Grace 2008). The interactions between different trophic groups on ecosystem functioning are hard to predict, as the functional effect of either trophic group in isolation can widely vary under the influence of a second trophic group (reviewed in Duffy et al. 2007). Despite the multiplicity of biotic attributes with a known functional effect, multi-trophic approaches to address the role of diversity on ecosystem functioning are extremely scarce. Rather, studies rarely extend beyond manipulations of two trophic groups within mesocosms, and are virtually nonexistent in drylands (Soliveres et al. unpublished data). The most comprehensive manipulations in drylands to date consist of the exclusion of either entire trophic groups (mainly mammals) or the exclusion of animals differing in body size (reviewed in Meserve et al. 2015). These studies have provided great insights into the dominance of top-down vs. bottom-up ecosystem control, and shifts in ecosystem organization derived from climatic changes in drylands. However, entire trophic groups are unlikely to go extinct, and it is more likely that large body-sized, specialist species disappear in response to climatic or biotic disturbances (see Naeem et al. 1994, Petchey et al. 1999, Duffy et al. 2007). Thus, understanding the functional implications of these more

realistic species losses can better inform about the consequences of biodiversity loss in drylands.

Supplemental Appendix 3. Grazing effects on soil carbon content and indices of ecosystem functioning

Data acquisition and handling

We used the ISI Web of Knowledge (www.isiwebknowledge.com) database (1945-2013 period) and the keywords “grazing” and “Australia” to extract data from published and unpublished reports, articles and reviews on the effects of European livestock grazing on plant, soil and animal variables. We compiled a database of 7621 records of an effect of grazing on 294 biotic and abiotic response variables from 224 studies (see Eldridge et al. 2016 for more details).

Many studies reported results for more than one response variables (e.g. plant biomass, plant richness, soil carbon, shrub cover), or the experiment was conducted at more than one independent location. In these cases, each contrast between any two levels of grazing, for a given response variable or case study, provided us with a separate measure of grazing effect size, but each was labelled by the particular study in order to account for the non-independence of measures within a study (see *Statistical analyses* below). We retained all measures from any one study as separate observations to ensure that our results were as general as possible (Maestre et al. 2005, Gómez-Aparicio 2009, Piñeiro et al. 2013, García-Palacios et al. 2013). Although this reduces the overall heterogeneity when estimating effect sizes, excluding multiple results from one data source can underestimate such sizes (Gurevitch & Hedges 1999).

Quantifying herbivore effects

From each of the studies we extracted quantitative and/or qualitative information on the level of grazing used in the study. This allowed us to place grazing intensity into four possible intensity categories: ungrazed, low, moderate or high grazing (see Eldridge et al. 2016 for details). We adopted the authors' own assessment of grazing intensity provided in a study because we acknowledged that they were best placed to describe the level of grazing at a particular site. The validity of this method has been tested previously (Eldridge et al. 2016).

For all of our attributes, we calculated an effect size for all the possible contrasts between the four levels of grazing (ungrazed, low, moderate, high). This resulted in six possible contrasts (ungrazed *vs* low, ungrazed *vs* moderate, ungrazed *vs* high, low *vs* moderate, low *vs* high and moderate *vs* high) ranging from the lowest contrast to the most extreme. We estimated an effect size as the natural logarithm (ln) of the response ratio (RR), i.e. $\ln RR = \ln (X_L/X_H)$ where X_L is the mean value of the response variable at the lowest level of grazing and X_H is that value for the highest level of grazing (Hedges et al. 1999). The log response ratio is negative when the value of a given response variable is lower as a result of a greater level of grazing. For example, if one study reported total soil carbon contents at ungrazed, low and high levels of grazing, we were able to calculate a log response ratio for three independent grazing contrasts for total soil carbon, i.e. ungrazed *vs* low, ungrazed *vs* high, and low *vs* high.

When the mean values of any record were zero (e.g. if the plant cover for an ungrazed record was 10% and that for a heavily grazed comparison 0%), we added to each of these values the minimum value that was likely to be detected with the sampling method used. Thus the ungrazed record would become 11% and the heavily grazed value 1% (Poore et al. 2012). This allowed us to improve our ability to detect useful effects of grazing on some response variables with infrequent or low values. Examination of funnel plots of effect sizes

vs. sample size did not indicate any publication biases that would be expected in cases of underreporting of non-significant results with low replication (Møller & Jennions 2001). Consistent with several recent meta-analyses (e.g. Mooney et al. 2010, Eldridge et al. 2016), we took the conservative approach of not weighting effect sizes by their variance.

The database used yielded 4902 possible comparisons of two levels of grazing. We extracted from this database information on the attributes that were related to soil carbon (total, labile and organic soil C contents; 184 comparisons), and three measures of ecosystem function ($n = 38$ comparisons each) that define the capacity of the soil to 1) resist disturbance (stability index), 2) infiltrate water (infiltration index) and 3) cycle nutrients (nutrient index; Tongway 1995, Tongway & Hindley 2004). These indices have been shown to be highly correlated with ecosystem functions related to soil stability, nutrient cycling and infiltration in dryland ecosystems worldwide (Palmer et al. 2001, Tongway & Hindley 2004, Ata Rezaei et al. 2006, Maestre & Puche 2009, Mayor & Bautista 2012). The three indices are assessed using rigorous, field-based protocols that assess the status and morphology of the soil surface within small quadrats (Tongway 1995, Tongway & Hindley 2004). Within these quadrats, 12 attributes (surface roughness, crust resistance, crust brokenness, crust stability, the percent cover soil erosion, deposited material and biocrust, plant basal cover, litter cover, litter origin, and the degree of litter incorporation, and soil texture) are measured (see Tongway 1995 and Tongway & Hindley 2004 for specific procedures).

Data analyses

We used linear mixed models in R (*lme 4*, Bates et al. 2014) with lnRR as the dependent variable, to examine the effects of increasing levels of grazing on the log response ratios for nitrogen and carbon, and the three functional measures (stability, infiltration and nutrient indices). The significance of these models was tested with likelihood ratio tests. Estimates of

lnRR were derived from REML and 95% bootstrapped confidence intervals for the estimates obtained from the likelihood profile.

Supplemental Appendix 4. Woody encroachment effects on synthetic ecosystem response variables

Data acquisition, handling and analyses

We used data derived from a systematic search of the scientific literature, described in Eldridge et al. (2011, 2012), to identify quantitative evidence of the extent to which the traits of woody species and aridity influenced the response of a range of ecosystem attributes to a change from grassland to woodland ('woody encroachment', see Eldridge et al 2011 for more details). We supplemented data published in Eldridge et al (2012) with 13 additional studies published since 2011 (Belay et al. 2013, Delgado-Balbuena et al. 2013, Gómez-Rey et al. 2013, Peng et al. 2013, Quero et al. 2013, Throop et al. 2013, Wiezik et al. 2013, Peng et al. 2014, Puttock et al. 2014, Scott et al. 2014, Seamster et al. 2014, Moreno-de las Heras et al. 2015). Together we derived a database of an effect of woody encroachment on 119 different ecosystem response variables that were derived from 174 studies conducted in 19 countries. To determine the effects of encroachment on our variables, we used the log response ratio, $\ln RR = \ln(E/U)$ for every variable, where E and U are the response variable in the plot with and without woody plants, respectively (Hedges et al. 1999). For example, the log response ratio for a plot that reported an increase in soil carbon from 0.5% to 1.5% after woody encroachment was equal to 1.099. We calculated 1822 separate comparisons of an effect of encroachment by 65 different woody species that were identified as encroachers (Supplemental Table 4), on a total of 119 response variables.

Classification and Regression trees

We analyzed the joint and independent effects of aridity and plant traits using Classification and Regression Trees (CART, De´Ath & Fabricius 2000). Rather than emphasize the relative importance of factors using probability tests, this approach actually produces a predictive model in the form of a decision tree. When CART is applied, one response variable is analyzed at a time. CART uses an algorithm to repeatedly partition the data into increasingly homogenous groups (or nodes), based on one predictor at a time. Subsequent splits can be made based upon the same or a different factor, increasing the number of nodes. Each end node is associated with a predicted mean value and standard deviation for the response variable, based on the samples falling within that node. We based optimal tree size on the sample-size corrected Akaike Information Criterion (AICc), by iteratively growing the tree by splitting to find the number of splits which minimized AICc. If a simpler model existed with an AICc value within 2 units of the minimum, the simpler model was selected; otherwise, the model with the minimal AICc was selected. To determine the model performance, we used a 10-fold cross-validation with 80% of data applied as training data, and 20% applied as validation data. This allows us to estimate the proportion of variance explained in data not used to construct the model (cross-validation R^2), an index of predictive ability of the model. We also report the proportion of variance explained in the data which was used to construct the model (training R^2).

As predictors, we used qualitative and quantitative information on ten structural (morphological) traits of the 65 woody species identified as encroachers. Our data were 1) mean height and 2) maximum height of encroaching plants at maturity (quantitative), 3) dispersal mode (wind, water or animal), 5) plant shape (rounded, v-shaped, weeping), 6) ground contact (yes, no), 7) allelopathic (yes, no), 8) palatability to mammalian herbivores (yes, no), 9) N-fixation ability (yes, no), 10) tap root (yes, no), 11) association with banded vegetation patterns (yes, no), deciduousness (yes, no). See Eldridge et al. (2011, 2012) for a

detailed explanation of the data sources used for obtaining this information. We also extracted data on aridity, determined as 1-Aridity Index [AI], where $AI = \frac{\text{precipitation}}{\text{potential evapotranspiration}}$ (UNEP 1992), and used it as an additional predictor. Data of the AI were obtained from the global aridity map of the FAO (<http://www.fao.org/geonetwork/srv/en/main.home?uuid=221072ae-2090-48a1-be6f-5a88f061431a>). Increasing values of aridity correspond with drier environments (see section below).

We developed five synthetic ecosystem responses as dependent variables in our models. Using our database of 1822 comparisons, we selected those individual attributes that provide a measure of ecosystem response in relation to: 1) ecosystem carbon, 2) ecosystem nitrogen, 3) soil hydrological function, 4) vascular plant richness, and 5) aboveground plant biomass. The attributes used to develop the five synthetic ecosystem response variables are shown in Supplemental Table 5. We constructed separate models for each response variable; i.e. carbon, nitrogen, soil hydrological function, plant richness and plant productivity models. Synthetic response variables, such as those described above, have been used extensively in both aquatic (index of biotic integrity, Karr 1991) and terrestrial (terrestrial index of ecological integrity, Andreasen 2001) systems. The approach we used is therefore multi-scale, flexible, measurable and comprehensive, and considers both the functionality of the ecosystem and community structural components of grassland-shrubland states (Eldridge et al. 2011). Using this system, one can integrate over any number of attributes, and the overall effect is largely independent of the conclusions drawn from the individual studies. CART analyses were conducted using JMP Pro 12 (SAS, Cary, NC, USA).

Supplemental Appendix 5. Relative importance of grazing, aridity and species richness as drivers of ecosystem functioning

The criteria for inclusion of study cases for our re-analyses were that studies were i) performed in the field, ii) conducted in drylands (areas with $AI < 0.65$), iii) included either aridity and grazing treatments, or observational data across gradients in aridity and grazing, iv) contained information on aridity or rainfall levels, grazing pressure, plant species richness/diversity and one or more ecosystem functions. Studies accounting for all these variables included plant biomass/ productivity (see details at the end of the section) and/or soil C as response ecosystem functions and, thus, we only included these in our analyses. Four different studies, including five independent study cases, fulfilled these criteria. These were databases gathered at the global (EPES-BIOCOM project; Maestre et al. 2012, $n = 131$), continental (Australia; Eldridge et al. 2016, $n = 64$ for plant biomass and 52 for soil C), regional (Patagonian rangelands; Gaitán et al. 2014, $n = 229$) and local scales (South African rangelands; Fynn 1998, $n = 60$ for rangelands in both good and poor condition). We searched for more study cases including all the information we required, both in the ISI Web of Knowledge and in Google Scholar using the combinations of keywords: "arid*" OR "dry*" + "herbivory" OR "grazing" + "function*" + "richness" OR "biodiversity". Some other studies accounted for variations in aridity and grazing, and contained measures of plant diversity and ecosystem functioning (China: Ren et al. 2012, Israel: Sternberg et al. 2015; USA and South Africa: Koerner & Collins 2014); however, we could not gather enough information from these studies to include them in our analyses.

To assess the relative importance of aridity, grazing and plant species richness as drivers of plant biomass and soil C, and to account for the well-known interrelationships between them, we used structural equation models (Grace 2006). Removal of plant tissues by grazing would be expected to reduce plant biomass and productivity, and this is the path

causality that we introduced. Additionally, plant biomass/productivity in drylands is mainly driven by water availability and soil fertility, which were introduced in our models whenever possible; thus we expect the co-variation between biomass and grazing to be accounted for by aridity. However, we cannot rule out the possibility that more productive sites can support a more livestock and therefore greater livestock pressure, and they could co-vary rather than be causally related. Indeed, this would explain the positive relationship between plant productivity and grazing found in the global (Maestre et al. 2012) and regional (Gaitán et al. 2014) studies.

Models were either saturated (and thus the overall goodness-of-fit could not be tested) or they showed a satisfactory goodness-of-fit (South African study cases; $\chi^2 < 1.2$; $P > 0.25$ in both cases). The latter suggest that the selected model structure provided a plausible fit to our data. To allow qualitative comparison between study cases we tried to fit a very similar *a priori* structure for all study cases, although we accounted for some particularities characterizing each database. Firstly, we accounted for temporal or spatial autocorrelation in our data by including rainfall from the previous year in those study cases including time series data (Fynn 1998) or both latitude and longitude in those with a nested structure (plots within countries [Maestre et al. 2012] or observations across different management or soil types [Gaitán et al. 2014]). The effects reported, therefore, are those after accounting for these potential confounding factors (but see full model output in Supplemental Figure 4). In those databases where both plant productivity and soil C data were available for the same study sites (Maestre et al. 2012, Gaitán et al. 2014), we included the correlation between them (see double-headed arrows in Figure 3 in the main text). Non-linear relationships were also evaluated when initial data exploration suggested this as a possibility (Eldridge et al. 2016). However, these non-linear relationships did not improve model fit, and thus were not included in the final models tested.

For some study cases (Fynn 1998, Eldridge et al. 2016) we could not gather aridity data but did have rainfall data; hence we used rainfall as our predictor but shifted the sign of the standardized path coefficients, as aridity level is inversely proportional to rainfall in drylands (Maestre et al. 2012). For the remaining study cases (Maestre et al. 2012; Gaitán et al. 2014), we obtained values of the AI from Zomer et al. (2008), who use the data interpolations provided by Worldclim (Hijmans et al. 2005). To facilitate the interpretation of results we calculated the aridity level of each site as $1 - \text{AI}$ so that higher values of this aridity level would indicate drier conditions (Delgado-Baquerizo et al. 2013). Due to the different methodologies and units used to estimate each variable, direct comparisons among study cases should be made with caution. Nevertheless, because the units were consistent within each study, and we calculated standardized path coefficients, we can qualitatively assess the spatial scale-dependency of the effect of each predictor (Supplemental Figure 5). To do this, we related changes in the standardized effect size of each predictor (aridity, grazing, plant richness) on plant biomass/productivity (as these were the attributes for which we had sufficient data) within each model to the spatial scale at which each study was performed (coded from 1 [global] to 4 [local]). The statistical significance of these relationships was tested using Spearman's rank correlations.

Data on grazing pressure were obtained using different methodologies in each database. For the global database (Maestre et al. 2012), grazing pressure was calculated as the stocking rate or sum of the number of livestock ($\text{animals} \cdot \text{km}^{-2}$) obtained from the FAO database of observed livestock density (http://www.fao.org/ag/againfo/resources/en/glw/GLW_dens.html). Other measures such as proportion of land dedicated to pasture or N deposition in manure (detailed in Delgado-Baquerizo et al. 2016b) were considered, but the former was related better to our surrogates of ecosystem functioning and thus is the one we used. For the Australian database, data were

directly obtained from each individual study and classified into four categories (see details in Eldridge et al. 2016 and *Quantifying herbivore effects* in Supplemental Appendix 3). Grazing data from Patagonian rangelands were obtained as a standardized measure of livestock stocking rate (kg ha^{-1}). This was done by obtaining from ranch managers information on ranch size and the average number of sheep, cows, goats and horses over the past five years. Livestock biomass was estimated as the product of the number of animals of each species and the individual average body weight, estimated using national statistics: 37 kg for sheep, 400 kg for cows, 17 kg for goats and 200 kg for horses (MECON 2002). Finally, grazing pressure in the South African local study cases was measured as number of grazing $\cdot \text{day}^{-1} \cdot \text{ha}^{-1}$ using information provided by field surveys and rangeland owners (see details in Fynn 1998). Methodologies for measuring above-ground plant biomass or productivity also differed among study cases. They ranged from plant productivity estimated from satellite estimates [NDVI] for the global database and the Patagonian rangelands (Gaitán et al. 2014, Delgado-Baquerizo et al. 2016a), to standing plant biomass based on quadrats harvested in the field or double sampling techniques in the Australian database (Eldridge et al. 2016), to the disc meter height method used in the South African study (Fynn 1998).

Supplemental Table 1. Spearman correlations (*P* values) between the diversity and abundance of bacteria and fungi, and between the diversity and relative abundance of some bacterial phyla, and different soil variables related to carbon, nitrogen and phosphorus cycling and availability (*n* = 78, except when noted with *, where *n* = 80). Bacterial and fungal abundance were log-transformed before analyses. *P* values below 0.05 are in bold. BAC = Bacteria, FUN = fungi, ACI = *Acidobacteria*, ACT = *Actinobacteria*, VER = *Verrumicrobia*, ASC = *Ascomycota*, BAS = *Basidiomycota*, PHO = phosphatase, BGL = β -glucosidase, NTR = potential N transformation rate, AVN = available N (ammonium + nitrate + dissolved organic N), PHE = phenols, ARO = aromatic compounds, HEX = hexoses, and PEN = pentoses; * *n* = 80. Original data come from Maestre et al. (2012, 2015).

	Shannon diversity index (bits)							Abundance (DNA copies·g ⁻¹ soil)		Relative abundance (%)	
	BAC	FUN*	ACI	ACT	VER	ASC	BAS	BAC*	FUN*	ACI	VER
PHO	0.405 (<0.001)	0.511 (<0.001)	0.238 (0.036)	0.246 (0.030)	0.395 (<0.001)	0.492 (<0.001)	0.340 (0.002)	0.351 (0.001)	0.367 (0.001)	0.533 (<0.001)	0.695 (<0.001)
BGL	0.362 (0.001)	0.399 (<0.001)	0.284 (0.012)	0.229 (0.044)	0.443 (<0.001)	0.359 (0.001)	0.148 (0.191)	0.579 (<0.001)	0.610 (<0.001)	0.355 (0.001)	0.414 (<0.001)
NTR	0.351 (0.002)	0.202 (0.072)	0.212 (0.054)	0.275 (0.015)	0.291 (0.010)	0.243 (0.030)	0.103 (0.361)	0.455 (<0.001)	0.470 (<0.001)	0.271 (0.016)	0.266 (0.019)
AVN	0.143 (0.211)	0.409 (<0.001)	0.120 (0.294)	0.033 (0.772)	0.265 (0.019)	0.353 (0.001)	0.217 (0.053)	0.265 (0.017)	0.373 (0.001)	0.195 (0.086)	0.241 (0.034)
PHE	0.421 (<0.001)	0.010 (0.931)	0.352 (0.002)	0.314 (0.005)	0.372 (0.001)	0.026 (0.828)	-0.029 (0.801)	0.356 (0.001)	0.472 (<0.001)	0.237 (0.036)	0.035 (0.762)
ARO	0.362 (0.001)	0.015 (0.896)	0.416 (<0.001)	0.352 (0.002)	0.271 (0.017)	0.031 (0.782)	-0.045 (0.694)	0.299 (0.007)	0.361 (0.001)	0.206 (0.070)	-0.004 (0.970)
HEX	0.474 (<0.001)	0.265 (0.018)	0.336 (0.003)	0.439 (<0.001)	0.212 (0.062)	0.238 (0.034)	0.105 (0.356)	0.333 (0.003)	0.456 (<0.001)	0.206 (0.070)	0.235 (0.038)
PEN	0.202 (0.077)	-0.273 (0.014)	0.183 (0.109)	0.155 (0.174)	0.304 (0.007)	-0.259 (0.020)	-0.072 (0.527)	0.207 (0.065)	0.342 (0.002)	0.073 (0.526)	-0.089 (0.438)

Supplemental Table 2. Spearman correlations (ρ) between nitrogen (N) deposition (sum of inorganic and organic N), N in manure and N in fertilizers and multiple ecosystem variables (plant diversity, total plant cover, microbial abundance, fungal-to-bacterial ratio, soil organic matter, soil spatial heterogeneity (i.e. organic matter heterogeneity), phosphorus availability and extracellular enzyme activities in global drylands (224 sites from all continents except Antarctica, describes in Maestre et al. 2012). Microbial abundance was estimated using quantitative PCR. Note that $n = 224$ for all variables except for microbial variables, were $n = 78$ (a subset of the sites surveyed in Maestre et al. 2012).

Ecosystem variables	N in manure ^a		N in fertilizer ^a		N deposition ^b	
	ρ	P	ρ	P	ρ	P
Plant diversity (Shannon index) ^c	-0.222	0.001	-0.147	0.031	-0.071	0.295
Total plant cover ^c	0.080	0.238	0.269	<0.001	0.276	<0.001
Soil organic C ^c	0.283	<0.001	0.548	<0.001	0.574	<0.001
Available P ^c	-0.256	<0.001	-0.238	<0.001	-0.323	<0.001
Available N ^c	0.284	<0.001	0.442	<0.001	0.175	<0.001
Activity of phosphatase ^c	0.192	0.005	0.367	<0.001	0.276	<0.001
Abundance of bacteria ^d	0.361	<0.001	0.311	0.007	0.388	<0.001
Abundance of fungi ^d	0.387	<0.001	0.632	<0.001	0.519	<0.001
Fungal: bacterial ratio ^d	0.144	0.214	0.493	<0.001	0.193	0.091

Data from: ^aDentener et al. (2006), ^bPotter et al. (2011), ^cMaestre et al. (2012), ^dMaestre et al. (2015)

Supplemental Table 3. The proportion of explained variance attributable to each predictor (trait or aridity), and estimates of total explained variance in the regression trees conducted with our five ecosystem response variables. Unless otherwise indicated (number in subscript), the predictor was used in only one split of the regression tree.

Aridity value and plant traits	Above-ground biomass	Carbon	Nitrogen	Soil hydrological function	Plant richness
Aridity	0.24 ²	0.32 ⁴	0.10	0.57 ²	
Height (mean)		0.39	0.66 ³	0.06	0.49 ²
Shape	0.11	0.05		0.06	
Palatability	0.09	0.02	0.09		
Deciduousness	0.02	0.05	0.11		
Rooting structure	0.08		0.05		
Banding	0.05				
Allelopathy	0.14				
Dispersal	0.26 ²			0.23	
Height (max)		0.08		0.09	
N-fixation		0.13 ²			0.51
Ground contact					
R^2 cross-validation	0.35	0.22	0.19	0.40	0.18
R^2 training	0.22	0.16	0.13	0.24	0.02

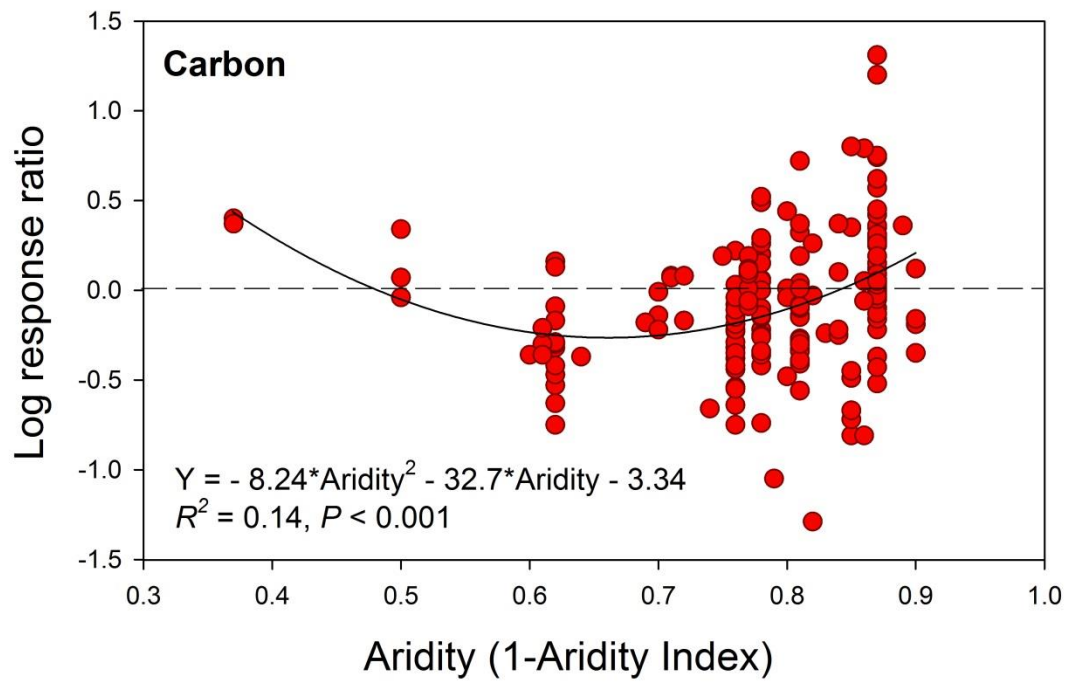
Supplemental Table 4. Woody species (shrubs and trees) identified as major encroachers in the 174 studies synthesized.

<i>Acacia aneura</i>	<i>Grewia flava</i>
<i>Acacia drepanolobium</i>	<i>Isocoma veneta</i>
<i>Acacia mellifera</i>	<i>Juniperus excelsa</i>
<i>Acacia saligna</i>	<i>Juniperus monosperma</i>
<i>Acacia tortilis</i>	<i>Juniperus occidentalis</i>
<i>Artemisia frigida</i>	<i>Juniperus osteosperma</i>
<i>Artemisia ordosica</i>	<i>Juniperus oxycedrus</i> subsp. <i>oxycedrus</i>
<i>Artemisia rothrockii</i>	<i>Juniperus virginiana</i>
<i>Artemisia tridentata</i>	<i>Larrea tridentata</i>
<i>Baccharis pilularis</i>	<i>Maerva crassifolia</i>
<i>Betula glandulosa</i>	<i>Mimosa biuncifera</i>
<i>Brachystegia spiciformis</i>	<i>Mulinum spinosum</i>
<i>Callitris glaucophylla</i>	<i>Myrica cerifera</i>
<i>Calluna vulgaris</i>	<i>Pinus ponderosa</i>
<i>Caragana brachypoda</i>	<i>Populus tremuloides</i>
<i>Caragana microphylla</i>	<i>Portulacaria afra</i>
<i>Chuquiraga avellanadae</i>	<i>Prosopis glandulosa</i>
<i>Cistus ladanifer</i>	<i>Prosopis laevigata</i>
<i>Cistus salviifolius</i>	<i>Prosopis nigra</i>
<i>Condalia microphylla</i>	<i>Prosopis velutina</i>
<i>Cornus drummondii</i>	<i>Prunus spinose</i>
<i>Curatella americana</i>	<i>Pteronia incana</i>
<i>Cytisus scoparius</i>	<i>Quercus coccifera</i>
<i>Dodonaea viscosa</i>	<i>Quercus ellipsoidalis</i>
<i>Eremophila mitchellii</i>	<i>Quercus emoryi</i>
<i>Eremophila sturtii</i>	<i>Quercus ilex</i>
<i>Eucalyptus coolabah</i>	<i>Retama sphaerocarpa</i>
<i>Eucalyptus crebera</i>	<i>Rosmarinus officinalis</i>
<i>Eucalyptus largiflorens</i>	<i>Senna artemisioides</i>
<i>Eucalyptus populnea</i>	<i>Shepherdia argentea</i>
<i>Flourensia cernua</i>	<i>Symphoricarpos occidentalis</i>
<i>Galenia africana</i>	<i>Terminalia sericea</i>
<i>Genista hirsuta</i>	

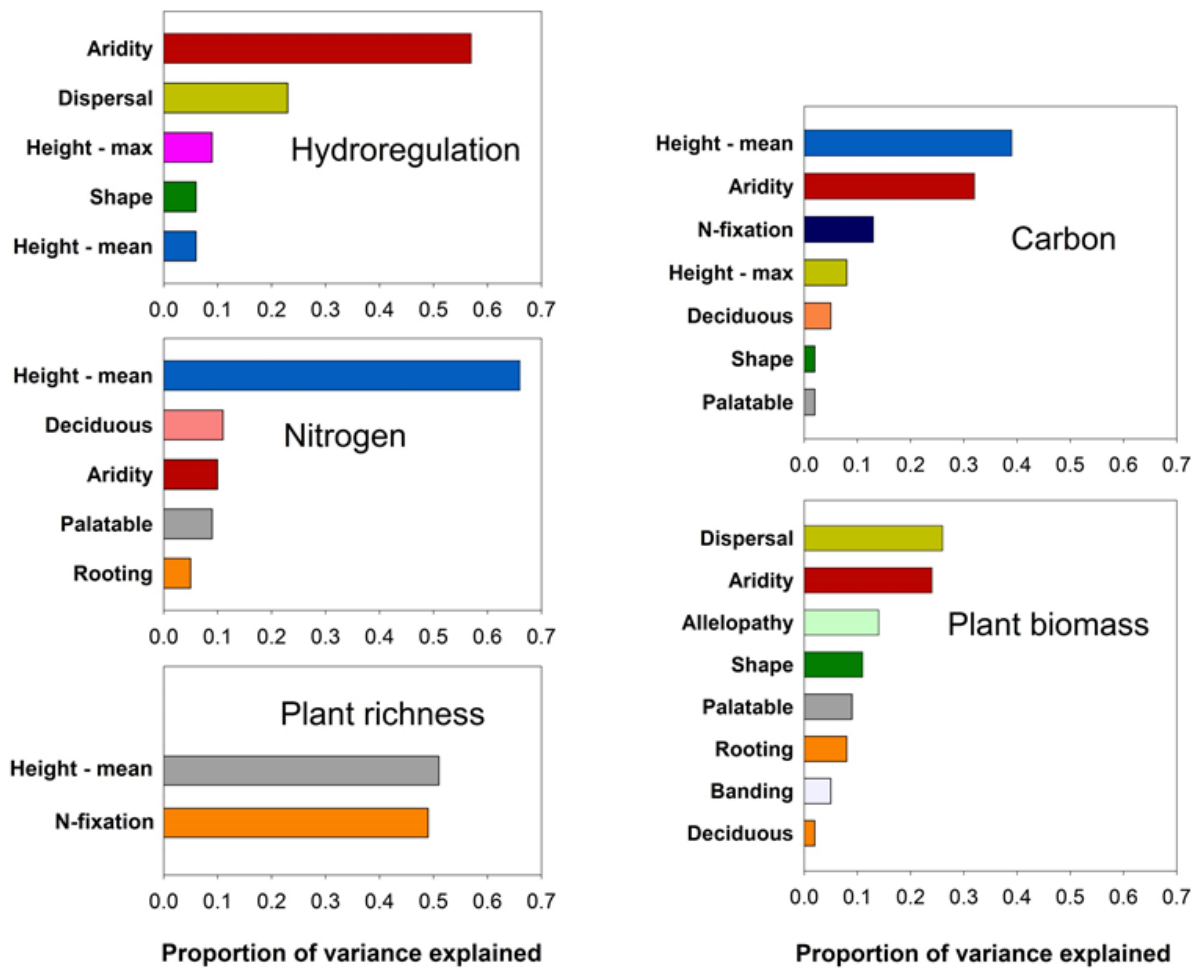
Supplemental Table 5. Specific attributes, their units of measurement and specific notes used to develop the five ecosystem response variables.

Ecosystem response variables	Attributes	Units	Notes
Carbon	Above-ground plant C	g/m ² or g/m ² /yr	Total plant carbon measured by combustion
Carbon	Microbial biomass	mg mic C/100 g	
Carbon	Microbial biomass C	mg/g	
Carbon	Soil inorganic C	%	
Carbon	Soil labile C	mg/g	KMnO ₄ reduction method (Weil et al. 2003)
Carbon	Soil organic C	%	Walkely-Black method
Carbon	Soil respiration	umol/m ² /s	
Carbon	Soil total C	%	Total C on combustion e.g. LECO method
Soil hydrological function	Discharge	L/min	
Soil hydrological function	Infiltration	mm/h	General measure of water flow through the soil
Soil hydrological function	Infiltration index	%	LFA infiltration index of Tongway (1995)
Soil hydrological function	Infiltration ponding	mm/h	Generally assessed with disk permeameter at +10 mm tension
Soil hydrological function	Infiltration tension	mm/h	Generally assessed with disk permeameter at -40 mm tension
Soil hydrological function	Runoff coefficient	%	Run-off as a percentage of total rainfall
Soil hydrological function	Runoff rate	mm/h	Steady-state rate of run-off
Soil hydrological function	Sediment concentration	g/L	
Soil hydrological function	Sorptivity ponding	mm/h ^{0.5}	Early infiltration phase under ponded (+ 10 mm) conditions
Soil hydrological function	Sorptivity tension	mm/h ^{0.5}	Early infiltration phase under tension (- 40 mm) conditions
Soil hydrological function	Volumetric moisture	%	Soil Volumetric moisture content
Soil hydrological function	Wetting front	mm/h	Depth to soil wetting front after rainfall
Nitrogen	Above-ground N	g/m ²	
Nitrogen	Microbial biomass N	g/m ²	
Nitrogen	Net soil ammonification	μg/g/day	

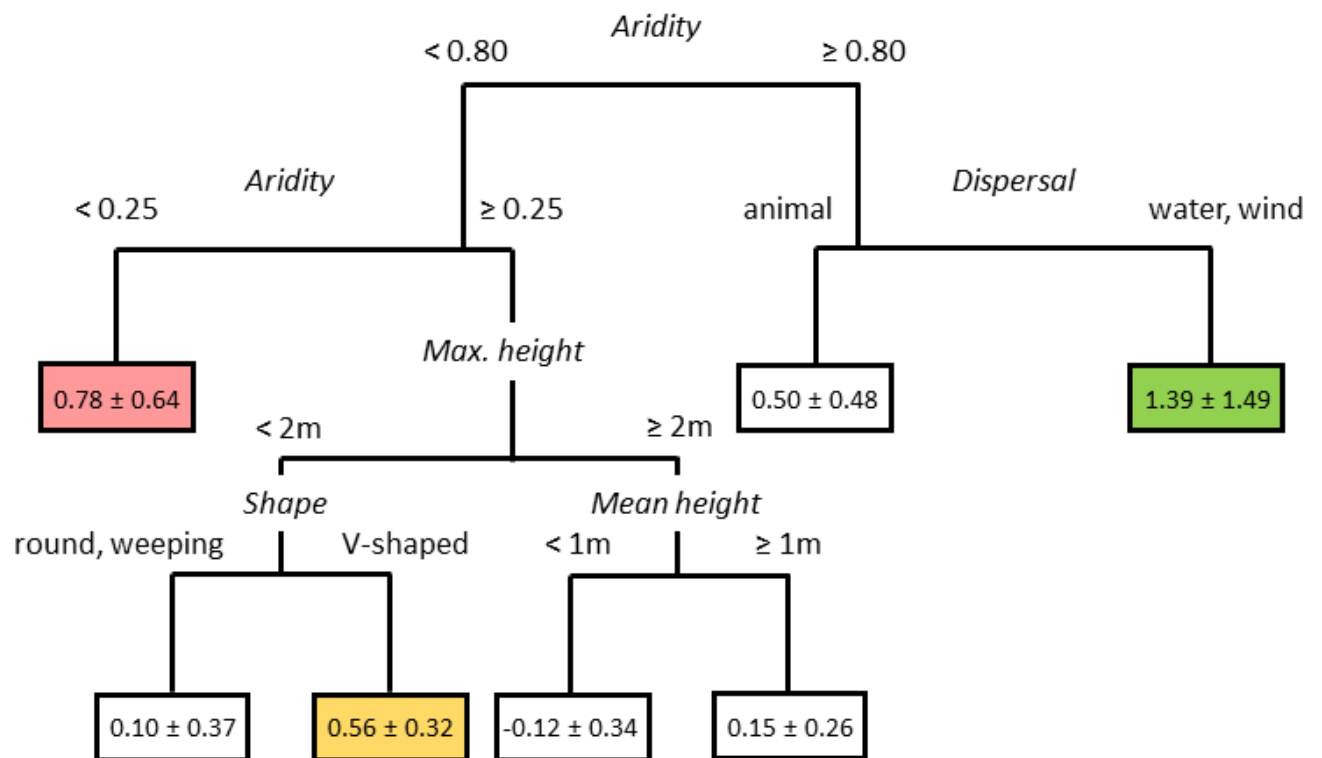
Nitrogen	Net soil nitrification	µg/g/day	
Nitrogen	Nutrient index	%	LFA nutrient index of Tongway (1995)
Nitrogen	Potential net soil N mineralisation	µg/g/yr	
Nitrogen	Shrub leaf N	g/m ²	
Nitrogen	Soil ammonium	mg/g	
Nitrogen	Soil available NH ₄ ⁺	mg/g or %	
Nitrogen	Soil available NO ₃ ⁻	mg/g or %	
Nitrogen	Soil inorganic N	%	
Nitrogen	Soil nitrate	mg/g or %	
Nitrogen	Soil organic matter	%	
Nitrogen	Soil total N	%	Generally assessed with total combustion (e.g. LECO)
Plant biomass	Above-ground ANPP	g/m ² /yr	Aboveground plant net primary productivity
Plant biomass	Above-ground biomass	g/m ²	Generally one-off assessment of plant biomass
Plant biomass	Grass root biomass	g/m ²	
Plant biomass	Litter biomass	g/m ²	
Plant biomass	NPP	g/m ²	Above and below ground net primary productivity
Plant biomass	Root biomass	%	Belowground net primary productivity
Plant biomass	Shoot biomass	g	
Plant biomass	Shrub biomass	g/m ²	Aboveground biomass of shrubs
Plant biomass	Shrub root biomass	g/m ²	
Plant biomass	Woody biomass	kg/ha	Aboveground biomass of trees and shrubs
Plant richness	Grass richness	# species	Aboveground biomass of grasses
Plant richness	Plant richness	# species	Richness (α-diversity) of all plant groups
Plant richness	Shrub richness	# species	Richness (α-diversity) of shrubs (woody plants < 3 m tall)
Plant richness	Tree richness	# species	Richness (α-diversity) of woody plants > 3 m tall
Plant richness	Woody richness	# species	Richness (α-diversity) of trees and shrubs



Supplemental Figure 1. Estimates of the log response ratio for soil carbon in relation to the increasing aridity (i.e. increasingly drier conditions). The average log response ratio for carbon across all grazing contrasts declined with increasing values of the aridity in environments from dry subhumid (Aridity: 0.3 – 0.4) to semiarid (Aridity: ~0.6).

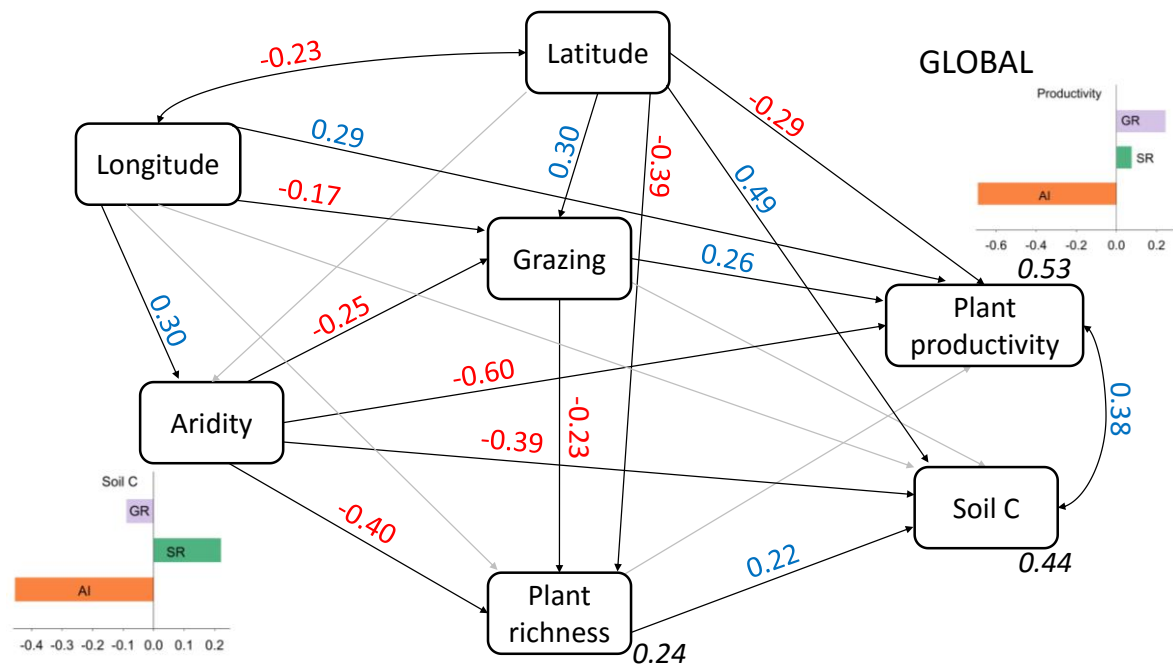


Supplemental Figure 2. The proportion of the variance in the log response ratio of the five ecosystem response variables evaluated explained by woody plant traits and aridity. Hydroregulation = soil hydrological function.

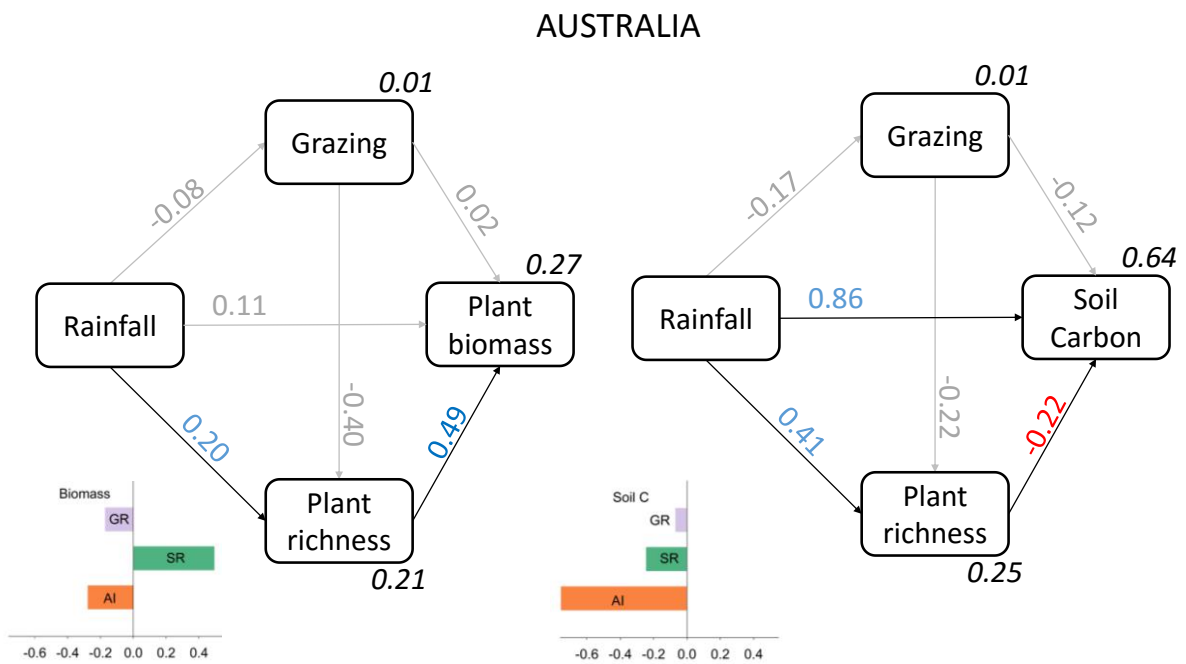


Supplemental Figure 3. Regression tree of woody encroachment effects (log response ratio, LnRR) on the soil hydrological function. Values in the boxes represent the mean (\pm SD) of the LnRR for soil hydrological function. In this model the largest LnRR value is highlighted in green. The values in pink and brown represent the largest LnRR values in the most arid and intermediate aridity areas, respectively. The cross-validation $R^2 = 0.24$, the training $R^2 = 0.39$.

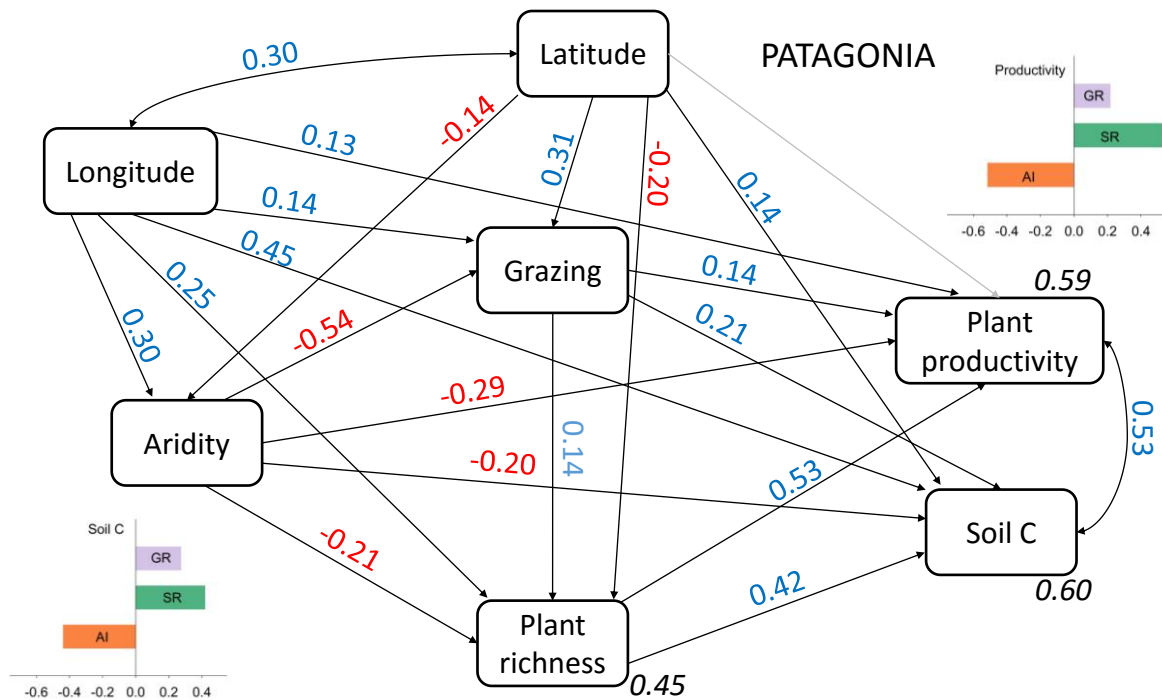
A)



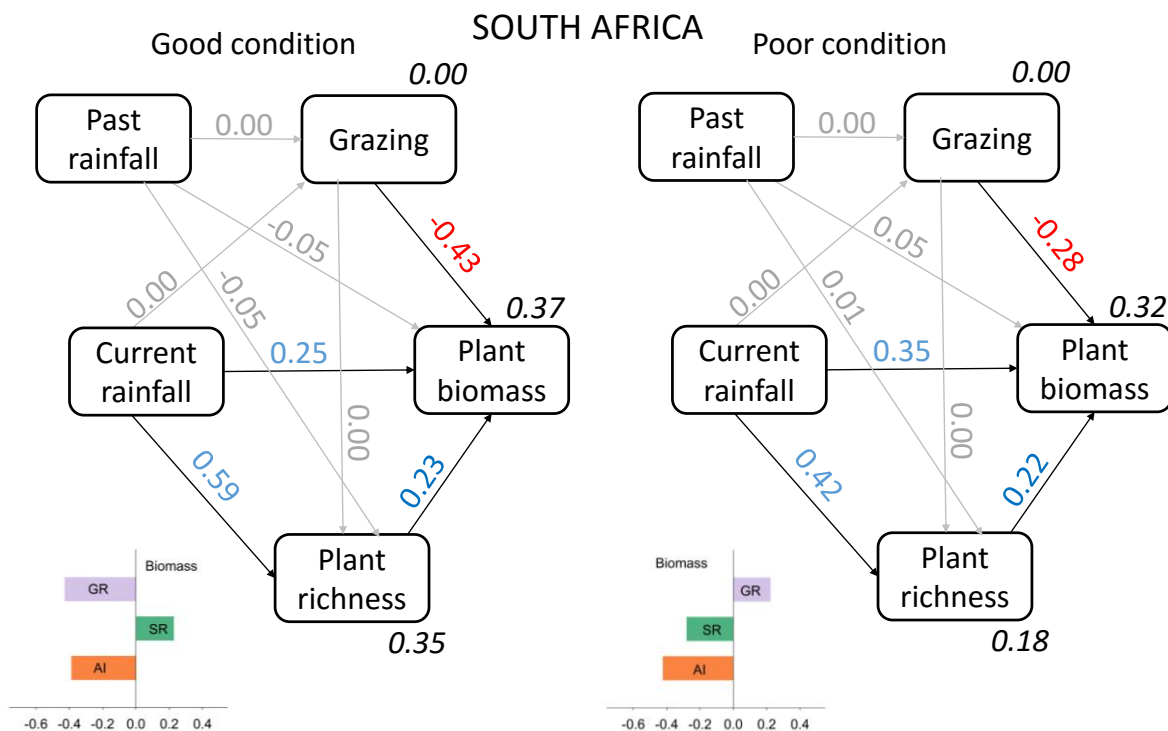
B)



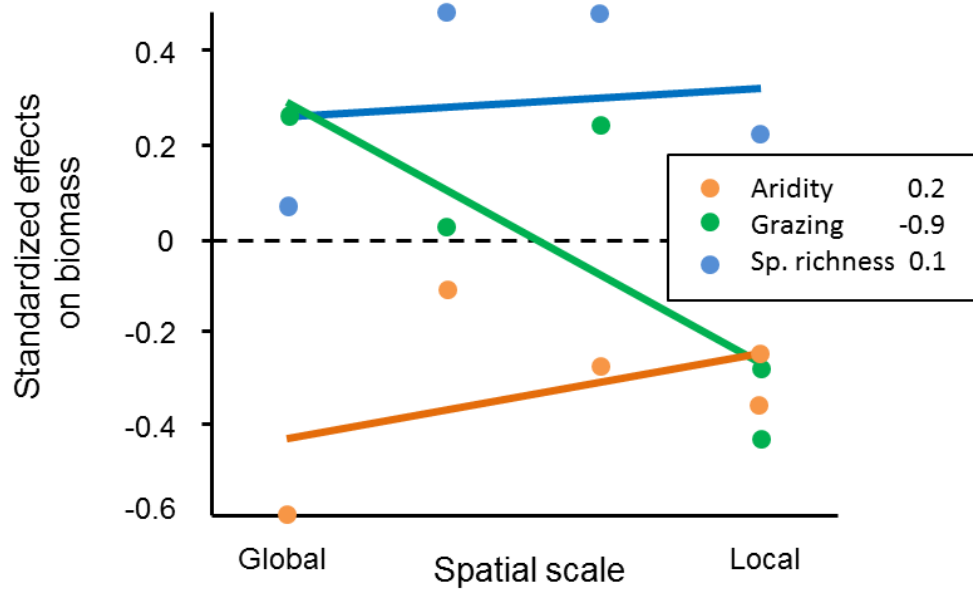
C)



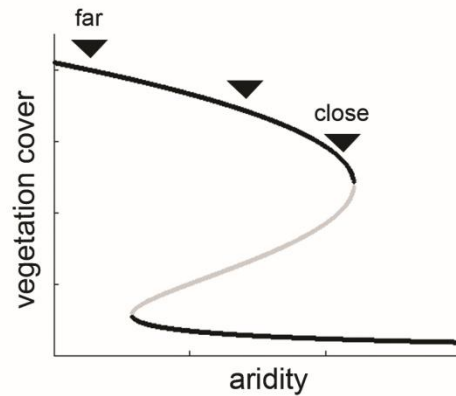
D)



Supplemental Figure 4. Detailed results of the SEMs presented in Figure 3 in the main text for the global (A), Australian (B), Patagonian (C) and South African (D) datasets. Non-significant, but tested, paths, appear in grey. For simplicity, all paths are the same size, with the coefficients indicating path size and sign (blue = positive, red = negative). Standardized total effects of grazing (GR; purple), plant species richness (SR; green) and aridity (AI, orange) are shown in the inset bar diagrams.



Supplemental Figure 5. Standardized direct effects on biomass/productivity change with the spatial scale at which each study case was performed (global, continental, regional or local), Spearman's correlation coefficients and regression lines are provided to aid interpretation.



Periodic vegetation patterns



Scale-free vegetation patterns



Supplemental Figure 6. Typical sequences of vegetation patterns observed along a degradation gradient (e.g. increasing aridity from left to right). Top: vegetation cover (in black) as a function of aridity. There is a range of aridity levels for which two possible vegetation covers are possible (one high and one low). Those two alternative stable states are separated by an unstable state (in grey) which delimits the attraction basins of the two stable states. Intermediate and bottom: examples of vegetation patterns observed along the aridity gradient, for low (left), intermediate (center) and high (right) aridity levels. The periodic vegetation patterns were generated by the model presented in Kéfi et al. (2010). They represent gap, labyrinths and spots from left to right. Scale-free patterns were generated by the model presented in Kéfi et al. (2007).

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